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Environmental Effects of Increased Coal Utilization Ecological Effects of Gaseous Emissions from Coal Combustion

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Agency

Environmental Research
Laboratory
Corvallis OR 97330

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June 1978

Research and Development

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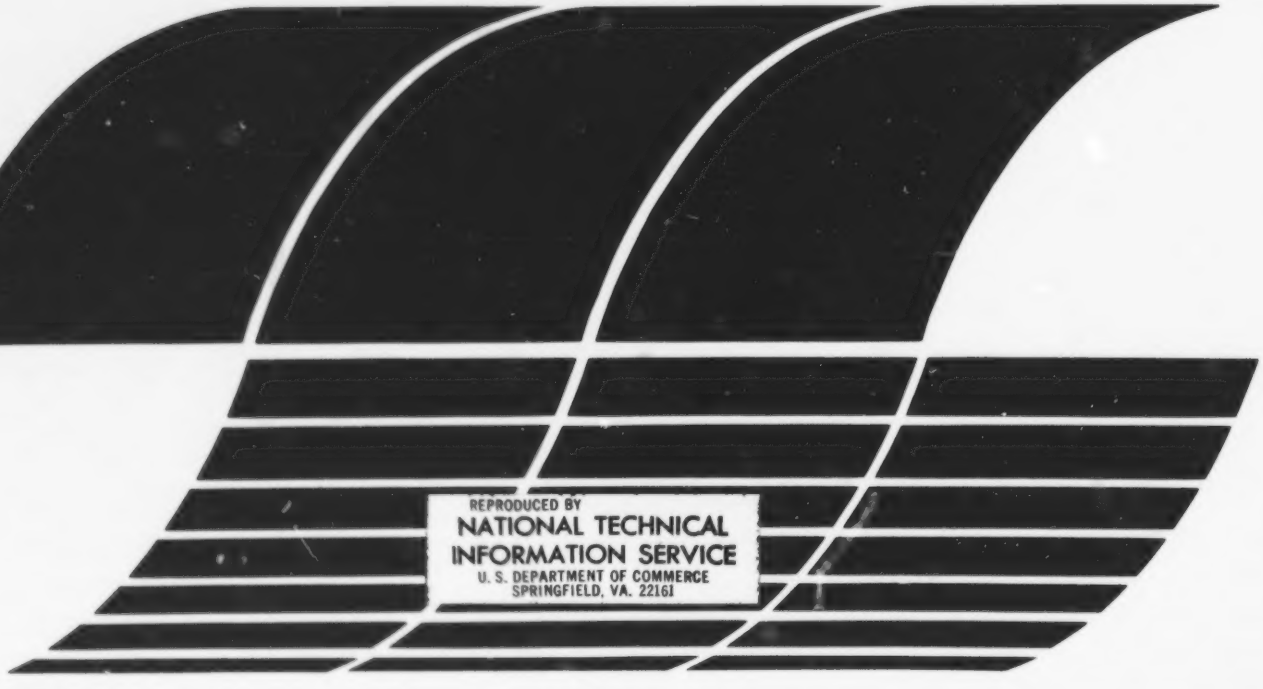
Environmental Effects of Increased Coal Utilization: Ecological Effects of Gaseous Emissions From Coal Combustion

Interagency Energy/Environment R&D Program Report



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16. ABSTRACT

This report is provided for the "Health and Environmental Effects of Coal Utilization" Committee (Dr. David Rall, Chairman) which was created by the request of the DOE in response to the President's Environmental Message. It evaluates ecological and environmental effects of gaseous emissions and aerosols of various types which result from coal combustion. The report deals with NO_x, SO_x. fine particulate, photochemical oxidant and acid precipitation as these pollutants affect natural and managed resources and ecosystems.

The economic implications of ecological effects are identified within acceptable limits In addition, the reliability of the data base upon which conclusions or estimates are made is evaluated to the degree possible. Aquatic and terrestrial effects are distinguished where the pollutants in question are clearly problems in both media.

Sulfur oxide (SO_x) emissions and nitrogen oxide (NO_x) emissions are projected to be higher in 1985 and 2000 than in 1975. Since SO_x and NO_x are major contributors to acid precipitation, substantial increases in total acid deposition can be expected in the nation as a whole. At present, acid precipitation is most abundant in the North Central and Northeastern States.

Estimates of the non health-related cost of air pollutants range from several hundred million dollars per year to $1.7 billion dollars per year. In general, these estimates include only those relatively easily measured considerations such as crop losses result-from acute pollution episodes or cost of frequent repainting as a result of air pollution.

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ENVIRONMENTAL EFFECTS OF INCREASED COAL UTILIZATION: ECOLOGICAL EFFECTS OF GASEOUS EMISSIONS FROM COAL COMBUSTION

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FOREWORD

Effective regulatory and enforcement actions by the Environmental Protection Agency would be virtually impossible without sound scientific data on pollutants and their impact on environmental stability and human health. Responsibility for building this data base has been assigned to EPA's Office of Research and Development and its 15 major field installations, one of which is the Corvallis Environmental Research Laboratory (CERL).

The primary mission of the Corvallis Laboratory is research on the effects of environmental pollutants on terrestrial, freshwater, and marine ecosystems; the behavior, effects and control of pollutants in lake systems; and the development of predictive models on the movement of pollutants in the biosphere.

This report further contributes to the knowledge of the EPA and will serve to provide ecological effects information on which policy and possible regulatory decisions can be based.

ABSTRACT

This report is provided for the "Health and Environmental Effects of Coal Utilization" Committee (Dr. David Rall, Chairman) which was created by the request of the Department of Energy in response to the President's Environmental Message. The paper is limited to an evaluation of the ecological and environmental effects of gaseous emissions and aerosols of various types which result from coal combustion. Throughout the report we will be dealing with NO_x, SO_x, fine particulate, photochemical oxidant and acid precipitation as these pollutants affect natural and managed resources and ecosystems.

In addition, synergistic effects involving two or more pollutants are evaluated as well as ecosystem level effects of gaseous pollutants. Also, we briefly summarize the effects on materials and atmospheric visibility of increased coal combustion.

The ordering of pollutants into the categories mentioned above has been done for the following reason. NO_x, SO_x and fine particulate are all primary emissions which are proximate to the power plant. Oxidant and acid precipitation are pollutants which generally develop at distances from the power plant itself and are involved in a long-range transport. Interactions or synergistic effects as well as the ecosystem level effects involve the whole system concept of evaluating the effects of air pollution and, hence, build upon the information developed in the earlier sections on NO_x, SO_x etc. The sequence in which these materials are dealt with is determined by the distance from the source as well as the degree of atmospheric transport and transformation that the materials must undergo. Finally, the order in which pollutant groups are dealt with reflects the progression of evaluating single pollutant exposures and ends in evaluation of more complex or combined effects.

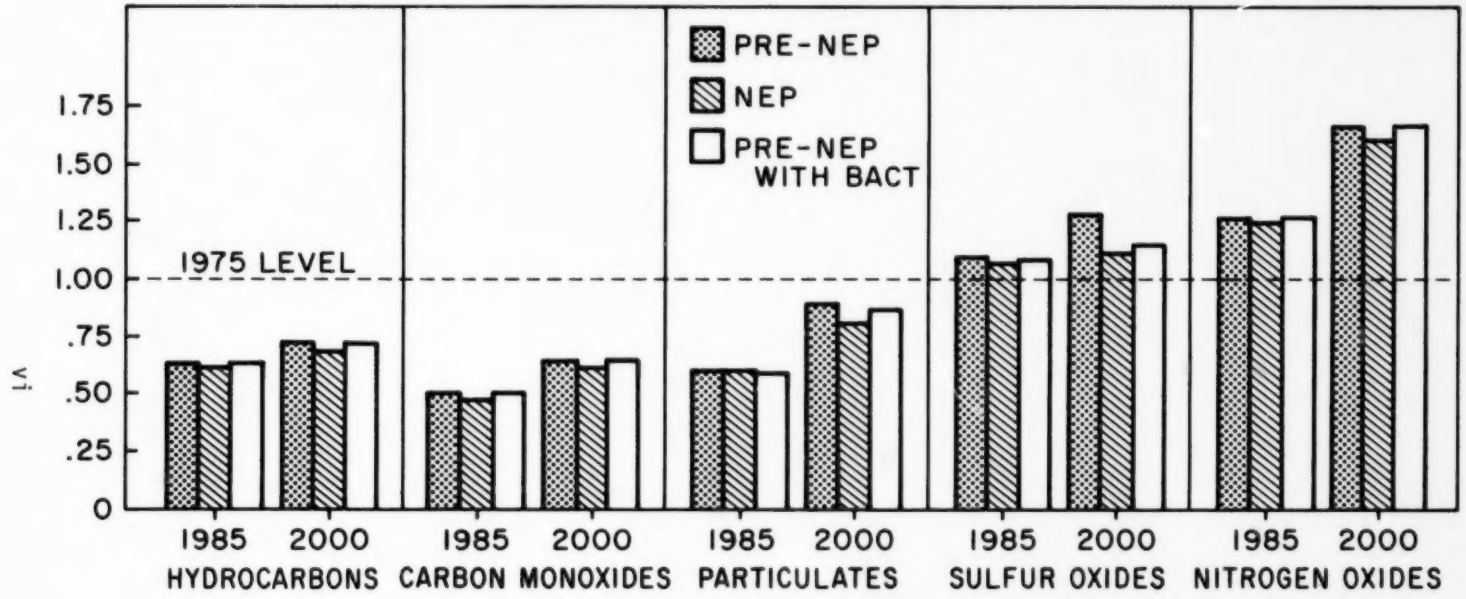
To the extent that they can be determined within acceptable limits, the economic implications of ecological effects are identified. In addition, the reliability of the data base upon which conclusions or estimates are made is evaluated to the degree possible. Both the short-term, high concentration, acute effects as well as low level concentration, chronic effects are addressed to the maximum possible extent. Aquatic and terrestrial effects are distinguished where the pollutants in question are clearly problems in both media. For example, since acid rain affects both air and water processes, we have a separate discussion of the aquatic impact of acid rainfall. However, most of the other pollutants, with the exception of fine particulate and fly ash, do not have a significant aquatic component and will, therefore, not be dealt with in the aquatic medium.

In order to assure that this report is on comparable grounds with the health effects of increased coal combustion as well as other related papers on increasing coal utilization, we have adopted the MITRE report (see Figure 1) as the starting point, or reference, for the levels of various pollutants which can be expected nationwide in 1985 and in the year 2000 in the U.S. both pre National Energy Plan (NEP) and assuming the National Energy Plan scenario. In addition, this starting point assumed that new source performance standards will be met by existing power plants as well as power plants under construction and that best available control technology will be used in retrofitting existing plants and in new power plant construction. Both of these assumptions can be seriously questioned. However, in the interest of using a uniform starting point, we have decided to use the MITRE data as the basis. (MITRE 1977, page 51). See Figure 1.

According to the estimates in the Mitre Corporation Report (MITRE, 1977), in both the National Energy Plan (NEP) and Pre-NEP scenarios, sulfur oxide (SO_x) emissions and nitrogen oxide (NO_x) emissions are projected to be higher in 1985 and 2000 than in 1975 -- SO_x about 12 percent higher and NO_x about 61 percent higher. Since SO_x and NO_x are major contributors to acid precipitation, substantial increases in total acid deposition can be expected in the nation as a whole. At present, acid precipitation is most abundant in the North Central and Northeastern States. Total SO_x and NO_x emissions are projected to remain high in these regions while increasing relatively more in the western than in the eastern regions of the country. The West's share of SO_x emissions increases from 22 to 29 percent while NO_x emissions increase from 31 to 39 percent. This means that the generally westward and southward spread of acid precipitation detected between 1955 and 1973 (see Figure 2) will continue. Considering that the U.S. is dominated by prevailing westerly winds, this means that total acid deposition in the eastern parts of the U.S. will probably increase still further.

As the MITRE Report indicates (p. 14): "In terms of gross emissions of particulates, SO_x and NO_x, coal is the least desirable fuel for utilities and industrial boilers. The heavy reliance on coal postulated in the NEP could have substantial adverse impacts on air quality in the absence of adequate control measures." Thus, any increase in combustion of coal relative to other fuels will tend to increase the seriousness of the acid rain problem unless total emissions are controlled.

A variety of ecological processes are affected and altered by air pollution, both gaseous and airborne particulates as well as aerosols. Such processes as community succession and retrogression, nutrient biogeochemical cycling, photosynthetic activity, primary and secondary productivity, species diversity and community stability, community respiration, dominant or abundance of key species within a community, and a variety of other attributes and processes may all be affected by pollutants (cf. Glass and Tingey, 1975; Dochinger and Seliga, 1976a, 1976b; Glass, et. al. 1977 for more complete discussion, literature review and citations). These ecological processes may be mediated or caused by alteration in a number of physiological processes or mechanisms or by population changes which may occur in the dominant or otherwise important species in a particular ecosystem.



AIR RESIDUALS LEVELS FOR 1985, 2000 RELATIVE TO 1975

Figure 1

Estimates of the public welfare (non health-related) cost of air pollutants range from several hundred million dollars per year to $1.7 billion dollars per year. In general, these estimates include only those relatively easily measured considerations such as the known losses to cultivated crops from acute air pollution episodes or the cost of frequent repainting required as a result of air pollution (Waddell, 1974). No substantial nationwide estimates of losses to forest productivity, natural ecosystem productivity which is tapped by domestic grazing animals and wildlife, and other significant dollar losses are available. This is partly due to the fact that such measurements are very difficult to make. However, the dollar losses which have been put forward as the consequence of air pollution damage are generally considered to be underestimates. In particular, dollar losses incurred from long-term, chronic, low-level exposure of crops, forests, and natural ecosystems to air pollutants remain viturally unmeasured nationwide.

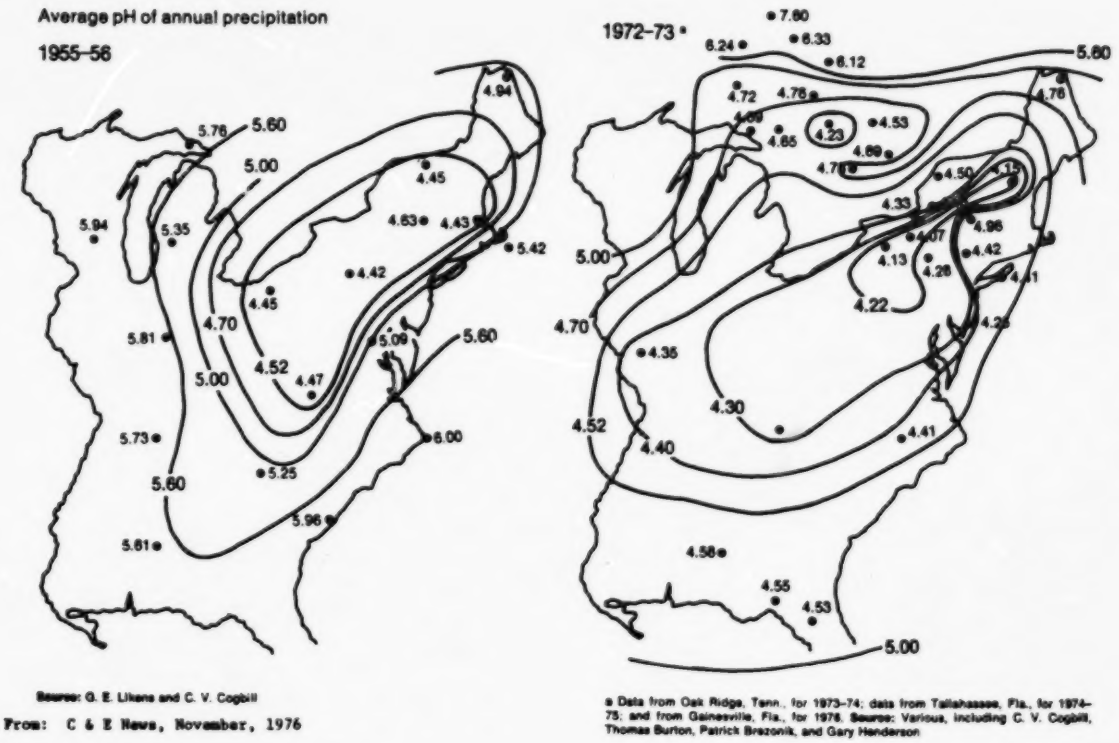


Figure 2. Time course of change in pH isopleths from 1955-56 to 1972-73 in the northeastern United States.

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Norman R. Glass

ECOLOGICAL EFFECTS OF THE OXIDES OF SULFUR COMPLEX

Airborne oxides of sulfur are emitted primarily from combustion of fossil-fuels, but other important sources include smelting of ores, production of sulfuric acid, and other industrial processes. Two basic approaches have been used to study the effects of sulfur oxides. One has been to monitor their occurrence at different sites around a source and determine what environmental changes are associated with these exposures. The other has been to expose plants or other receptors to known dosages under controlled conditons and thereby derive a dose-response relationship and the effects of various factors upon it. The goals of all these studies have been to develop methods for the identification and estimation of the effects of air-borne sulfur oxides, to understand their mode of action, and provide adequate criteria for air quality standards. An understanding of the effects of SO_x in the biosphere must consider that they are phytotoxic, interact with other pollutants, and that sulfur in proper amounts is essential for all organisms.

Effects of Sulfur Oxides

Vegetation

Sulfur dioxide penetrates the leaf through stomates. In the aqueous phases of the foliar tissue it forms bisulfite or sulfite ions, which are then oxidized to sulfate. The sulfur derived from SO_2 enters the sulfur pool, is converted to sulfur-containing compounds, is translocated throughout the plant, and ultimately passes to the soil through exudation from roots, elution from leaves, or leaching from litter. Sulfur may also be released directly to the atmosphere from the leaf by the light-mediated conversion to H_2S (de Cormis, 1969).

The reactions of sulfite or its oxidation products with membranes, enzymes, or other cellular components can produce altered metabolic functions and changes in the ultrastructure of the cells. If sufficiently great in rate or extent of occurrence, these changes lead to a loss of chlorophyll, disruption, or death of the cell. The effects of sulfur dioxide at the cellular level may also be manifested as a decrease or increase in stomatal aperture. Through the death or disfunction of foliar tissue or the inhibition of gaseous exchange, photosynthetic fixation of CO_2 by the plant is reduced, permanently or temporarily. The gradual accumulation of sulfate in the foliar tissue may reach toxic levels and lead to injury or premature senescence and defoliation. The net effect of reduced photosynthesis is altered growth and reproduction. Changes in the inorganic and organic constitutents of foliage have also been noted (EPA, 1973; Rennie and Halstead, 1977).

Many factors are known to affect the response of plants to atmospheric sulfur dioxide. The major biological factors known to affect tolerance to SO_2 are species, genetic background, and a stage of development. For crop species, the median tolerances span a 10- to 20-fold range of dosages, and the same range of tolerance may be present in natural populations. There is also considerable variation within any one species, and cultivars of certain crops or selections of forest trees have been rated with respect to SO_2 tolerance. When leaves are fully-developed they appear to be most susceptible to injury by SO_2; the foliage of conifers and evergreens appears to be most tolerant during winter and periods of low temperature that depress metabolic activity. Moreover, the effects of foliar injury on yield appear to be greatest at one stage of development in the plant for crops such as beet, cotton, or soybean (Van Haut and Stratmann, 1970).

Environmental factors significantly affect SO_2 tolerance and plants are most susceptible when exposures occur at temperatures above 5-6°C, relative humidities above 75%, in periods of higher light intensity, and with adequate soil moisture: i.e., under conditions that most favor gaseous exchange by the foliage. Adequate supply of mineral nutrients, particularly nitrogen and to some extent phosphorus, calcium, and potassium, tends to increase the tolerance of trees and crops to SO_2.

Taking into account both biologic and environmental factors, the degree of risk of SO_2 to various kinds of cropland use have been estimated: conifers, pome fruits, and berries are at greatest risk whereas cole crops, potato, beet, and some ornamentals are at least risk (Guderian, 1977).

Animals

Toxicology studies of sulfur oxides in laboratory animals indicate that sulfuric acid aerosol is more toxic than gaseous SO_2 (HEW, 1970). A consideration of more recent data (summarized in EPRI, 1976) indicates that acute exposures of mammalian systems to SO_2 require higher dosages to produce deleterious effects than are necessary for the production of acute adverse effects in vegetation. An indirect action of SO_2, through a change in the suitability of plant, litter, or soil as a habitat, may be of more significance than a direct effect on invertebrates and microfauna (Halstead and Rennie, 1977). The same may also be true for vertebrates.

Soils

The interaction between soils and air-borne oxides of sulfur has received relatively little attention. The two major effects of SO_2 pollution of the soil are a decrease in pH and an increase in sulfate (Halstead and Rennie, 1977). A change in these two variables may then affect the structural and microbial characteristics of the soil. In some systems where the soils are sulfur deficient or where structure is improved by the addition of sulfur, increased productivity may result. However, these circumstances are relatively rare. When the rate of addition or the amount added is excessive, changes in the composition, structure, and function may result. Any effect of SO_x on vegetation will indirectly affect the soil. A reduction in vegetative canopy allows greater solar radiation to penetrate to the soil and the

reduction in litter accumulation leads to greater extremes in temperature, less retention of moisture, and a greatly altered microclimate in the upper soil horizon.

Other Effects

Microorganisms are essential for the functioning of terrestrial and aquatic ecosystems through their roles in primary production, nitrogen fixation, and in decomposition processes. The impacts of air pollution on these organisms have not been studied extensively. Recent work suggests that adverse effects on photosynthetic carbon fixation can be induced in blue-green algae by low concentrations of SO_2 (applied as bisulfite). Other types of algae were relatively resistant. Respiration of bacteria and protozoa indicated that these groups are also relatively tolerant (Wodzinski et al., 1977).

Indirect Effects

Indirect effects of SO_x include changes in the occurrence of pathogens and destructive insects. SO_x in combination with other gaseous and particulate pollution has resulted in additive, synergistic and antagonistic responses among certain insect pests and some pathogenic organisms. (see section on interactions).

Tolerance to SO_2 in spruce has been associated with resistance to other stresses (Guderian, 1977). Consequently, genetic shifts and changes in community diversity may result from long-term exposures.

Magnitude of the Effect

Recent Canadian estimates(Rennie and Halstead, 1977) of the magnitude of the problem in Canada indicate that 6.5×10^6 metric tons of SO_2 are emitted annually; areas affected adversely cover between 1.1 to 2.5×10^4 km^2; and direct economic losses to forestry lie between \$1.2 and \$$2.8 \times 10^6$. A 1971 estimate (Benedict, et al. 1971) of the annual losses in the United States owing to the effects of SO_2 on plants was \$$6.2 \times 10^6$; and pollution threatened areas were estimated to total about 13% of the land area of the USA in which 57% of the national population resided.

Both of the above studies indicate that uncertainties are present in their estimations as to the real extent and degree of economic effects. Moreover, both are of the opinion that the values given may be underestimates. Based upon projected small increases in SO_x emissions (Figure 1) there is little likelihood of significantly increased effects on vegetation. However, in areas directly influenced by SO_x emissions and/or in those areas where there is at present small margin of safety for vegetation injury the probability of increased risk to vegetation may be large.

Areas of Uncertainty

Perhaps one of the most uncertain and controversial problems is the relation between exposure and effect. The occurrence of foliar injury and aesthetic damage may largely be due to short-term, high-level exposures. At

least three quantitative formulations have been given for liminal exposures with time (duration of exposure) and dose-rate (atmospheric concentration) as variables, with changes in concentration being relatively more important than changes in time. One formulation also indicates the total dose needs to be evaluated also with respect to the frequency of exposure (Male et al., 1977). Effects on crop yield are not always related to foliar injury and the chronic as well as acute exposures over the course of a growing season must be weighed. For the estimation of adverse ecological effects, not only acute and chronic exposures but also total sulfur input to the system, as estimated by the annual mean loading, may be of significance.

Researchers need air-monitoring data that is sufficiently thorough to indicate the functions and range of parameters that adequately describe realistic exposures. It should be noted that the joint distribution of SO_2 and other air pollutants is also of concern. Furthermore, attention should be given to the occurrence of H_2SO_4 where meteorological conditions or cooling tower drift may promote its near-site occurrence.

The biological data base should also be expanded to include more information on the statistics that characterize the tolerance of natural populations, especially those of more arid regions. Most of our knowledge is derived from agricultural crops and species of temperate forests. This biological data base should also be expanded by a further inventory of effects, particularly at the ecosystem level which are of a cumulative and indirect nature, i.e., those that deal with the biotic interactions of pathogen or pest with host, competition and selection in communities and populations, and biogeochemical changes.

Particular attention should be given to the following problem areas:

(1) Dose-response: an acceptable model that relates these two variables is still needed. Adaptation and further development of air-surface transport models probably offer the most efficient route and best means of incorporating the effects of biological and environmental factors and transients. Further studies with controlled exposures may be needed to estimate model parameters.

(2) Thresholds: there is a need to elucidate the mechanisms and capacities of micro-(metabolic) and mega-(bio-geochemical) systems to fix atmospheric sulfur. Such knowledge is necessary not only to predict maximum tolerable concentrations in the short-term but also sorptive capacity of vegetative sinks for the long-term.

(3) Interactions: considerable work still needs to be done to give a reasonable description of the interactive effects of two or more pollutants in combination.

ECOLOGICAL EFFECTS OF THE OXIDES OF NITROGEN COMPLEX

The combustion of fossil fuels produces oxides of nitrogen; higher combustion efficiency results in increased production of this pollutant. The increase in fossil fuel consumption necessary to meet future national energy requirements, coupled with reductions in imported oil, increases in domestic coal utilization, and improved combustion technology, will result in greater emissions of nitrogen oxides (Figure 1).

Nitric oxide (NO) is the predominant oxide released in combustion. It enters the photolytic cycle and is converted to nitrogen dioxide (NO_2) and ultimately to other oxides of nitrogen. Of the various oxides of nitrogen, only NO and NO_2, collectively referred to as NO_x, are recognized as important air pollutants with respect to their potential adverse effects on living systems.

The effects of NO_x is necessarily complex because nitrogen is an element essential to all biological systems, it is toxic by itself, and it participates in the atmospheric formation of other air-borne pollutants.

Natural sources account for more than 90% of global oxides of nitrogen (NRC, 1977) and the most abundant of them, nitrous oxide (N_2O). Anaerobic bacterial action and chemical decomposition of nitrate are considered the major natural sources of NO. Forest fires and other uncontrolled combustions contribute only a small portion of atmospheric NO_x. However, even though natural sources of NO_x are large compared with manmade sources, there are two considerations which make manmade sources of critical importance. First, nitrous oxide constitutes over half of the world wide naturally emitted NO_x compounds, but it is not considered a pollutant because it is in low ambient concentrations and does not appear to be involved in the nitrogen dioxide photolytic (photochemical smog forming) cycle. Second, local concentrations of manmade NO_x are often large compared with background ambient levels and are involved as oxides precursors which enter into the photochemical reactions leading to air pollution.

In 1970 the two major classes of man-made sources of NO_x, mobile and stationary, contributed 51% and 46%, respectively, of the total NO_x emissions in the United States. Nearly 80% of the mobile sources of NO_x were from motor vehicles. NO_x emissions from electric power generation alone could increase from a 1972 base of 3.95×10^6 tons/year to 7.89 or 16.57×10^6 tons/year by 1990; the larger emission predicted assumes no enlargement in nuclear plant capacity (U.S. Senate, Committee on Public Works, 1975).

The occurrences of NO_x in concentrations that directly affect the most susceptible components of terrestrial ecosystems (vegetation) are not the result of emissions from mobile, or most stationary sources. Rather, direct effects of NO_x, where they occur, are confined to localized areas near industries that use or manufacture nitric acid, and are usually associated with accidental acute exposure.

Effects of Nitrogen Oxides

Vegetation

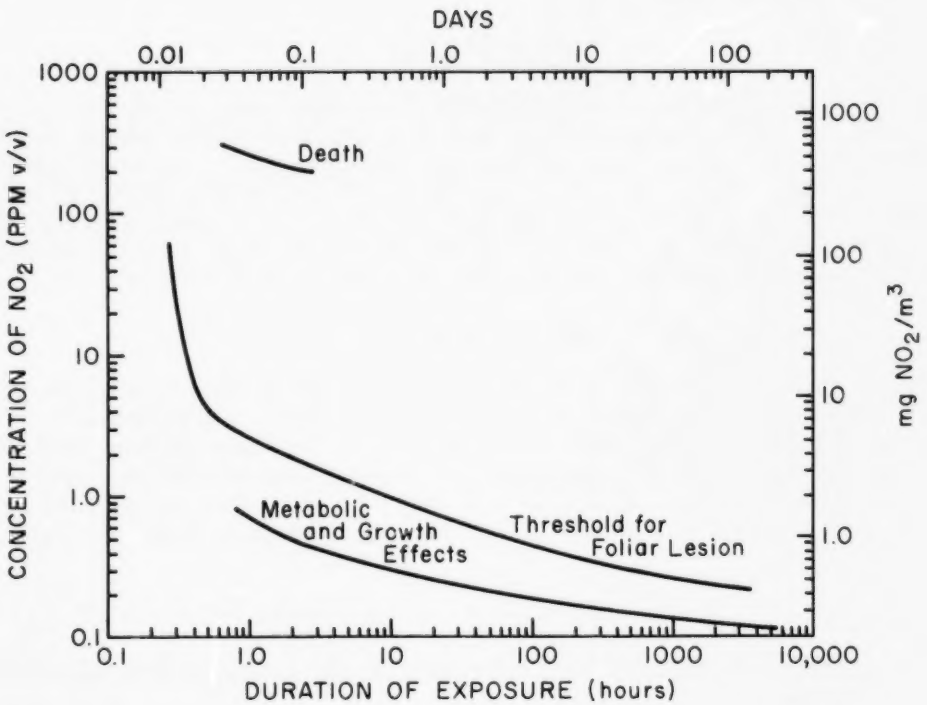
Nitrogen oxides are significantly less phytotoxic than other constituents of the photochemical oxidant complex, and their major importance in the atmosphere with respect to vegetation injury is an indirect one: i.e., participation in photochemical reactions resulting in the production of atmospheric oxidants, including both ozone and the peroxyacyl nitrates.

Two recent reviews (MacLean, 1975; NRC, 1977) presented an overall synthesis of NO_2 induced effects in plants. Atmospheric NO_2 is absorbed by plants primarily through stomata, whereupon it changes from a gaseous to an aqueous form. It alters the pH of the cellular milieu and reacts with cellular constituents leading to altered metabolism, ultrastructural changes, reduced photosynthesis, and probably many other effects that have not been observed or measured. These, in turn, lead to effects at progressively higher levels of biological organization. Premature senescence, chlorosis, necrosis, or abscission of leaves affect the entire plant causing reduced growth or reproduction, and even death of individual plants or entire plant communities.

Because of the relationship between the concentration of NO_2 and the duration of exposure, there is no single threshold dose for an effect in plants. Rather, threshold doses are usually described as functions of pollutant concentration and exposure time (Figure 3). The threshold curves in Figure 3 are approximations to indicate the NO_2 doses that result in: the death of plants; foliar lesions; and metabolic or growth effects. At NO_2 concentrations below the threshold for metabolic and growth effects, vegetation can serve as a sink to remove NO_2 from the atmosphere (Hill, 1971), and this absorbed NO_2 may be metabolized by the plant.

Animals

The aim of experimental exposures of animals to NO_x has been more to elucidate potential hazards to human health than to determine effects on animals *per se*. Experiments with animals that used extremely high concentrations of NO_2 for short periods or lower concentrations for very long periods have induced pulmonary and extrapulmonary effects and mortality. However, prolonged exposure to NO_2 concentrations of up to 0.5 ppm did not affect normal laboratory rats, mice, or rabbits (NRC, 1977). In short-term exposures (1-hour) the mortality threshold for those animal species was 40 to 50 ppm NO_2 (Hine *et al.*, 1970). However, there is no evidence that current levels of atmospheric NO_x, even those that occur in large urban areas (e.g., Los Angeles), have an effect on animal communities.


DAYS
0.01
0.1
1.0
10
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100
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1.0
0.1
CONCENTRATION OF NO_2 (PPM v/v)
Death
Metabolic
and Growth
Effects
Threshold for
Foliar Lesion
1000
100
10
1.0
mg NO_2/m^3
0.1
1.0
10
100
1000
10,000
DURATION OF EXPOSURE (hours)

Figure 3.

Soils

Studies to assess the relation between atmospheric NO_x and effects on soils are limited. Both NO and NO_2 react readily with soils and are usually converted to nitrate, and sorption of large amounts of NO_2 under experimental conditions decreases soil pH (Ghiorse and Alexander, 1976). Little is known of the effects of NO_x on the microfauna and microflora of the soil. Although these microorganisms are critical to the balance in terrestrial ecosystems, experimental evidence is limited to the responses of individual species *in vitro*. These experiments have demonstrated that high concentrations of NO_x affect growth or survival of individual microorganisms in defined media, but effects at NO_x concentrations likely to be encountered in the ambient atmosphere are unknown (NRC, 1977).

Other Effects

Because of the importance of microorganisms in primary production, nitrogen fixation, and decomposition processes, a reduction in these activities by air pollution could have serious consequences in terrestrial and aquatic ecosystems. Wodzinski *et al.* (1977) found profound effects on photosynthetic carbon fixation in blue-green algae by exposure to low concentrations of NO_2 (applied as nitrite). Other types of algae were relatively resistant, as was the rate of respiration in bacteria and protozoa.

Indirect Effects

Indirect effects would include the participation of NO_x in atmospheric reactions leading to the production of other, more toxic photochemical oxidants i.e. ozone and peroxyacyl nitrates and an increase in the rate of transport or the amount of nitrogen passing through the ecosystem. An example of indirect effects of NO_x is the combined influence of NO_x and one or more other air pollutants when they occur simultaneously (See section on Interactions).

Summary of Nitrogen Oxide Effects

1. Higher plants are the most susceptible receptors in the terrestrial ecosystem.

2. The indirect effects of NO_x (i.e., its role in the formation of ozone and peroxyacyl nitrates and the synergistic effects on plant injury of low concentrations of NO_2 and sulfur dioxide mixtures) are probably more important than direct effects.

3. NO_2 is more toxic than NO.

4. Limited studies with animals indicate that very high NO_2 concentrations are required for morbidity or mortality.

5. Soils absorb NO_x, which is converted to nitrate, but effects of NO_x on soil-borne microorganisms have not been investigated.

6. As a component of acid precipitation, NO_x is part of a pollutant complex which does exert substantial ecological impact as discussed under the acid precipitation section of this report.

Magnitude of the Effect

Atmospheric NO_x is relatively unimportant in its direct effect on ecosystems relative to other major pollutants. Concentrations of NO_x necessary to affect susceptible members of the terrestrial ecosystem (vegetation) are much greater than for sulfur dioxide and other photochemical oxidants. Estimates of the economic impact of air pollutants on vegetation (Benedict et al., 1971) did not include NO_x by itself; rather, it was combined with ozone and peroxyacetyl nitrate.

As noted above, injury to vegetation from atmospheric NO_2 is usually the result of accidental releases or spills of NO_x and it is localized. Widespread effects probably do not occur. For example, the average NO_2 concentrations in most major cities of the United States are well below the threshold for effects on metabolism and growth of plants (Figure 3). The maximum NO_2 concentrations recorded in Los Angeles (EPA, 1971) for an entire year (1966) were only slightly higher than the thresholds for averaging times of one day or less, and below the curve for longer sampling periods.

The real importance of NO_x, however, is probably its potential for indirect effects on environmental quality. The synergistic effects on plant injury of low concentrations of NO_2 and sulfur dioxide found in experimental exposures, and role of NO_x in the production of photochemical smog and acid precipitation are probably where NO_x poses the greatest threats. Even though NO_x emissions will increase substantially (Figure 1) one can not reliably estimate any change in the direct or indirect effects, based on our limited knowledge of NO_x effects.

Areas of Uncertainty

According to one estimate (MITRE, 1977), increases in nationwide emissions of NO_x will be 61% higher in 1985-2000 than they were in 1975. At local levels, near power plants for example, emissions will be even greater.

An understanding of the significance of an increase in NO_x emissions can best be realized by the development of models to predict the concentrations and dispersal of NO_x around power plants. This kind of information is necessary to design fumigation experiments in which the exposure regimes can be programmed to simulate those that are likely to occur. These models are not yet available.

The direct effects of NO_x become more relevant as emissions increase. Research should initially emphasize effects on vegetation because of its susceptibility relative to other classes of organisms. Since plants and other organisms are relatively insensitive to NO, research should concentrate

on NO_2. The gross effects of acute NO_2 exposures on metabolism, growth, and yield of crop plants and population changes in natural plant communities are unknown. The importance of soils and vegetation canopies as sinks for the removal of NO_x and its metabolic role is not well understood. Thus the possible effects of NO_x as a source of nitrogen in the biosphere have not received serious attention.

Laboratory exposures of animals should be carried out in parallel with field investigations to assess effects on population dynamics, behavior, and reproductive success. The effects of NO_x in combination with other gases and metals is poorly defined (see section on Interactions).

ECOLOGICAL EFFECTS OF THE PARTICULATE COMPLEX

Coal and oil fired electric power plants are among the largest anthropogenic point sources of particulate matter. Adoption of electrostatic precipitation for up to 98 percent of the particulates produced by coal combustion is rapidly reducing the particulate emissions from these sources. If new plants continue to follow best available control technology (BACT), and progress can be made in converting non-conforming plants to BACT, total particulates from power generation and heating could be reduced 30 percent or more by 1985 (MITRE 1977), in spite of greatly expanded coal utilization. The significance of the remaining particulates for visibility and biological function is still a serious question because of our incomplete understanding of particle chemistry, transport and interaction with the biota. Fortunately several excellent reviews are available on selected aspects of this topic. These include Berry and Wallace (1974), Vaughan et al. (1975), Van Hook and Schults (1977), and NRC (1977).

Size, Transport and Deposition

Particulate contaminants originating from coal combustion with best available control technology are released into the atmosphere in sizes ranging from 0.1 to 6 micrometers. In the absence of rainfall, size is of great importance because of the potential that particles of this size will remain airborne for days or weeks and be transported 100 to 1000 km. Other studies have shown that less than 5 percent of the particulates from large power generating facilities will reach the ground or be intercepted by vegetation within a 20 km radius. These emissions, therefore must be viewed as a regional or global problem.

The mechanics of deposition of particles on natural surfaces has been gleaned from studies with particles ranging from 1 to 50 micrometers, as reviewed by Chamberlain (1967) and Ingold (1971). Particulates can be deposited on natural surfaces by three processes: The largest by sedimentation under the influence of gravity but little of this size remains under BACT; impaction on obstacles to air flow (e.g., vegetation under the influence of eddy currents); and deposition under the influence of precipitation. For particles of the 0.1 to 3 micrometer size range, impaction is most important. It occurs when air flows past an obstacle and the airstream divides, but particles in the air tend to continue in a straight path due to their momentum and strike the obstacle. The efficiency of collection via impaction increases with decreasing diameter of the collection obstacle and increasing diameter of the particle.

Ingold (1971) presented data indicating that leaf petioles are considerable more efficient particulate impactors than either the twigs (stems) or the leaves of plants. For particles of dimensions 1-5 micrometers, interception of fine hairs on vegetation is possibly the most efficient retentive mechanism. The capturing efficiency of raindrops falls off very sharply for particles of 5 μm or less. Particulate removal by stomatal uptake has been suggested (Jordan, 1975) and photographed by Helmke et al. (1976), but it is infrequent. When it occurs, stomatal function fails.

Thus the surfaces of vegetation provide a major filtration and reaction surface for the transfer of fine particulate pollutants from the atmosphere to the biosphere. The capability of plants to act as a sink for air contaminants has been recently reviewed (Smith and Dochinger, 1976). The hypothesis that plants are important particulate traps is supported by evidence obtained from studies dealing with radioactive, trace element, pollen, spore, salt, precipitation, dust and unspecified particles.

Effects of Fine Particulates and Areas of Uncertainty

The significance of fine particulates for plants and animals derives from certain aspects of their physical presence and their chemical composition. Constituents with potential toxicity for plants include B, Cd, Co, Cr, Cu, F, Ni, Tl, and V. Constituents important for animals include Be, Cd, F, Hg, Ni, Sb, Se and Tl. The five trace elements with greatest potential for adverse impaction on the terrestrial biota, ranked in descending order of biological impact and research need include Cd, Ni, Tl, Cu and F (Van Hook and Schults, 1977). Of particular interest is the evidence suggesting that some trace elements are preferentially concentrated on the smallest particles (Linton et al., 1976). Some elements also are present in the gas phase (Natusch, 1975; Natusch and Wallace, 1974; Andren et al., 1975; Pellizzari et al., 1975; Jones et al., 1976). While enrichment of particles appears to be independent of plant operating conditions, stack gas temperature strongly influences the distribution of elements between phases.

Recently, much more concern is being expressed about the potential toxicity of polycyclic aromatic hydrocarbons (PAH) which are released in coal combustion and can be condensed on the fine particulates. Studies in Norway (Lunde et al., 1976) have shown significant quantities of these compounds on particulates collected from regional air samples over southern Norway. Since the PAH compounds are known to be active in metabolism (including enzyme induction) and generate a wide range of metabolites, research is beginning to focus on their importance as carcinogenic agents for all animals breathing fine particulates, and for aquatic organisms taking up these compounds from the surrounding water (Lech and Melancom et al., 1977). Current research indicates important effects induced in fish at dilutions of a few parts per billion, well below levels observed in the Norwegian rain-water samples, but the hazard for human use of the fish is still unknown. Since most particulates in the future will be in the fine aerosol size, the 30 percent reduction in tonnage of emissions given in Fig. 1 may or may not represent a reduction in ecological hazard, depending on activity of these organic compounds.

Evidence also is available to indicate that the soil component of terrestrial ecosystems is an important sink for fine particulates. Studies of the ecology of soil ecosystems are concerned with the potential interference that the above trace elements can have on the metabolism of microorganisms, arthropods and other soil animals.

Many of these organisms are involved in organic matter decomposition or mineralization. Subtle changes in these processes could have profound effects on ecosystem function by altering the quality or quantity of nutrient cycling (Bond et al., 1976; Lighthart and Bond, 1976).

Acute plant disease is infrequently ascribed to particulate contamination. Particulates, therefore, are generally not considered harmful to vegetation (Jacobson and Hill, 1970; U.S.D.A. Forest Service, 1973). In numerous and varied situations, however, particulates have been implicated in subtle adverse vegetation effects (Lerman and Darley, 1975).

Still another consequence of fine particulates is the reduction in photosynthesis occasioned by reduced light from atmospheric contamination (Czaja, 1966; Darley, 1966; Treshow, 1970). Reduced photosynthesis due to increased stomatal diffusion resistance caused by particulate plugging has also been proposed (Ricks and Williams, 1974). Slight tissue burning appears in electron micrographs of fly ash particles on leaves (Lech and Melancom, 1977), a suspected result of the organic compounds on the particles. Quantitative estimates of the importance of these effects are available.

Summary

1. The fine particulate materials emitted by coal-fired power generating plants using best available control technology can remain airborne for a considerable period of time; probably less than 5 percent of these materials will reach the ground or be intercepted by vegetation within a 20 km radius of the source.

2. Particles are deposited on natural surfaces by sedimentation, impaction, and in precipitation.

3. Plants are important sinks for particulate pollutants and the fine hairs present on the surface of plants are the most effective interceptors of these particles.

4. Particles containing heavy metals, other elements, and polycyclic organics may have adverse effects on terrestrial and aquatic biota.

5. The soil is also an important sink for fine particles and there is concern that the contamination of soils with heavy metals could alter the normal processes associated with microflora, microfauna and nutrient recycling.

6. Although fine particles are relatively non toxic to plants, fly ash has been reported to induce foliar lesions and reduction in photosynthesis because of reduced light or interference with stomatal function.

7. There are interrelationships between fine particles and the incidence and severity of infestation by certain insects and mites (see section on Interactions).

ECOLOGICAL EFFECTS OF THE PHOTOCHEMICAL OXIDANT COMPLEX

Photochemical oxidants, namely ozone (O_3) and to a much lesser extent, peroxyacyl nitrates (PANs), are the most damaging air pollutants affecting agriculture and forestry in the USA (Jacobson, 1977). Of the homologous family of PANs, peroxyacetyl nitrate (PAN) is most common in polluted atmospheres. It is more phytotoxic than O_3 but the ambient concentration of PAN is much lower than O_3 (Taylor, 1969). Formation of oxidants in the atmosphere has a complex dependence on amount of primary precursors (nitrogen dioxide and hydrocarbons) present, meteorological conditions and time of day (NAS, 1977). Therefore, it may be assumed that the substantial increase in nitrogen oxides (NO_x) predicted for 1985 and 2000 (Figure 1) suggests a concomitant increase in oxidants in coming years.

Plant Effects

Investigations in the early 1900's provided evidence that low concentrations of ozone could produce foliar lesions but the first observation of injury on field grown plants was reported in the early 1940's (Middleton et al., 1950). The photochemical oxidants have constituted a chronic problem in southern California for more than 30 years. The extent and severity of losses to agricultural crops, exclusive of forests, native vegetation and ornamental plantings, increased to $3 million in 1953 (Middleton et al., 1956) and by the 1970's crop damage in the state had reached more than $55 million (Millecan, 1976). Millecan also reported that 18 different crops could no longer be profitably grown in some areas of southern California. A crop loss of $5 million was reported for Connecticut in 1953 (Heggestad, 1966).

Chronic oxidant injury to ponderosa (Pinus ponderosa Laws) and Jeffrey pine (P. Jeffrey Griv. and Balf.), the dominant species comprising the mixed conifer forest ecosystem in the San Bernardino (SBNF) and Angeles National Forests (ANF), has been estimated using aerial photography. In the SBNF, 46,230 acres had heavy ozone type injury, 53,920 acres moderate injury and 60,800 acres light or no injury (WERT et al., 1971). Severity of injury to ponderosa and Jeffrey pines at selected plots in the SBNF generally increased between 1973 and 1975 and the mortality rate was as high as 6-8% (Miller et al., 1977).

Annual observations in the Cleveland National Forest of southern California showed increased injury from 1974 to 1976 (U.S. Forest Service, 1977).

A ground survey of parts of the Sequoia Natinal Forest (SNF) in 1974 revealed 24 locations where foliar symptoms of O_3 were present on one or more species (Williams et al., 1977). Oxidant injury to similar forest tree species has been identified in other locations in California (Miller and Millecan, 1971) but the most important examples are in the ANF, SBNF and SNF.

In the eastern United States "Emergence tipburn" of eastern white pine (P. strobus L.) is associated with elevated O_3 (Berry and Ripperton, 1963). Injury to new needles was induced in the field by 0.03 ppm O_3 for 48 hours and 0.07 ppm O_3 for 4 hours (Costonis and Sinclair, 1969). Morris (1973) found that 30% of the white pines in one area of Shenandoah Valley of Virginia developed injury while 5% were considered severely affected.

More important in the east, however, is the widespread damage to a wide variety of crops, particularly dry beans, grapes, potatoes, and tobacco. Moderate damage is reported on these and some other crops almost every year east of the Mississippi River and any increase in damage would have serious consequences economically.

Sources of Oxidant Precursors

The urban source of primary pollutants was as far as 80 to 160 Km upwind in each of the chronic oxidant injury incidents to forest vegetation cited above. The afternoon onshore flow through the "reactor" (Southcoast air basin) and upslope to the forested mountains is an example of a set of terrain and meteorological conditions which produce adverse oxidant concentrations in the forest (Edinger, 1973). An area source the size of metropolitan Los Angeles could cause ozone concentrations to exceed the federal standard of 0.08 ppm at locations as far away as 250 Km (Blumenthal et al., 1974) and the effect may carry over more than one day. At St. Louis the federal standard was exceeded at distances as far as 150 Km downwind (White et al., 1977). In the midwestern and eastern United States, high ozone concentrations in rural areas result from a high pressure system as it moves from west to east. A large system with a radius of 375 Km or more with high ozone may result as precursors are picked up over densely populated areas during the west to east movement (Ripperton et al., 1977).

Tesche, et al. (1977) and Hegg et al. (1977) concluded that no ozone was synthesized in the "near field" (out to 90 Km) of plumes from coal or gas fired plants but Davis et al., (1974) suggested that O_3 synthesis did occur close to such sources. In either case the potential for increased O_3 synthesis beyond the 90 Km point during the passage of a large high pressure system (Ripperton et al., 1977) is a distinct possibility.

Effects On Other Organisms

Although ozone and PAN effects are best known on vascular plants, several researchers have reported measurable effects of ozone at ambient concentrations on various phases in the life cycle of pathogenic organisms (NAS Report, 1977). Ozone and mixtures of ozone with sulfur dioxide were shown by one researcher to decrease the population of four nematodes associated with soybean (NAS Report, 1977).

Direct effects of ozone and PAN on arthropods have not been documented but oxidant (ozone) injury to ponderosa pine was shown to predispose trees to later invasion by pine bark beetles (Miller et al., 1977). The beetles increased the rate of decline and may be considered the final cause of tree death.

There is no documented evidence that free ranging mammals, birds and reptiles are directly affected by O_3 and PAN air pollutants but it is readily apparent that severe injury to a wide variety of vascular plants may drastically alter food availability and cover for such animals.

No direct effect of oxidant pollutants on aquatic habitat and soils has been established. Because of the strong oxidizing potential it has been assumed that the oxidants react at the surface of the soil and bodies of water and little if any penetration occurs. Soil and aquatic organisms are thus assumed to be protected against direct impact from the oxidants.

Areas of Uncertainty

Areas of uncertainty which may have an important bearing on the effect of photochemical oxidants on the ecosystem associated with the expansion of coal utilization are:

(1) What added injurious effects will result from simultaneous occurrence of other pollutants emitted by coal combustion? What effect will the projected increase in NO_x and decrease in hydrocarbon (MITRE, 1977) have on ambient oxidant levels? How will dose response relationships be altered by increased levels of CO_2?

(2) What are the relative stabilities of organism populations in different types of ecosystems? Will changes due to air pollutants be irreversible?

(3) Can criteria be developed to interpret or evaluate services rendered by ecosystems in addition to the value of harvestable products?

Summary

One of the most threatening aspects of the photochemical oxidant problem is the long range transport of plant damaging concentrations in urban plumes even at present levels of precursor emissions. During periods of high meteorological potential for air pollution the elevated ozone concentrations become a regional phenomenon in both the eastern and western United States. Repeated episodes during 4 to 6 months of the year are sufficient to cause obvious foliage injury, impaired photosynthesis and growth reductions of sensitive species populations, in both agricultural and forest ecosystems. During the last 20 to 30 years, the problem of elevated oxidant concentrations has become a new stress in several ecosystems particularly in the Southwest.

Energy policy options which have the potential for increasing NO_x emissions will surely affect the extent and the intensity of photochemical oxidant stress in ecosystems on both a local and regional scale. The kinds and number of changes to be expected in the biotic communities comprising affected ecosystems can only partially be anticipated with the present state of knowledge and methodology. The "goods and services" formerly provided by certain ecosystems may be in short supply or perhaps nearly eliminated. The costs in time and energy required to attempt conversion back to a condition that could accomodate former "demands" may be unacceptable even if methods were identified and tested.

ECOLOGICAL EFFECTS OF ACID PRECIPITATION

In addition to naturally occurring sulfur oxides and hydrogen sulfide, atmospheric loads of sulfur which lead to acidic precipitation in various regions are augmented by sulfur dioxide and particulate sulfate from a combination of fossil fuel combustion and from industrial processes. Further, nitrogen oxides also contribute to acid precipitation.

Sulfur and nitrogen compounds are removed from the atmosphere by two processes; a) dry deposition including the absorption of SO_2 on exposed surfaces and the sedimentation and impaction of particulates and b) wet deposition in which sulfur compounds are frequently deposited as acid precipitation. Acidity of precipitation should be understood as a reflection not only of the amounts of sulfuric, nitric, hydrochloric, and organic acids in the atmosphere, but also as a reflection of the balance between all the cations and anions dissolved in precipitation. Some of these ions include beneficial nutrients; other are injurious to plants and animals.

Dry deposition is a continuous process depending mainly on the concentration of sulfur oxides near the ground, the yearly amounts deposited generally decreasing with increasing distance from the source. Wet deposition is much more variable being dependent both on the rainfall pattern and on the burden of sulfur compounds within the mixing layer. It can be substantial in areas exposed to precipitation from air which has passed large emission sources. In cold climates air pollutants deposited during the winter usually accumulate in the snow pack. When this melts, much of the pollutant load is released in concentrated form with the first melt water. This may lead to sudden increases of acidity in the watercourses and also to some extent in the soil.

Strong acids, such as sulfuric, nitric and hydrochloric acid, have lowered the pH of rain and snow falling on much of eastern U.S. to between 3 and 5. Recent data indicate that in the U.S. there has been both a southwestward and westward extension of the region of acid precipitation and an intensification of acidity in the northeastern region since the mid-1950's. As shown earlier in Figure 2, precipitation in a large portion of the eastern United States was less than pH 5.6 in 1955-56; the zone of greatest acidity (lowest pH) was generally consistent with the zone where sulfur emissions were high. By 1973, however, the region of low pH embraces most of the area east of the Mississippi River.

Currently, the pH of precipitation in much of the northeastern U.S. averages between 4.0 and 4.2 annually. But values between 2.1 and 3.6 have been observed for individual storms at various locations (Likens and Bormann, 1974 and 1975; Likens, 1976) in most cases many hundreds of kilometers from major sources of air pollution.

Similar pH patterns have been reported in northwestern Europe. For example, in Norway the maximum wet deposition of pollutants occurs in the southern part of the country, and the mean pH is below 4.3.

Data from New York State and New England indicate that about 60 - 70% of the acidity is due to sulfuric acid (Galloway et al., 1976; Likens, 1976). Along with the important role of sulfur in affecting acidity, studies in New York State and New Hampshire indicate that the relative importance of nitric acid has increased during the last 10 years (Likens, 1976).

Effects of acid precipitation on aquatic ecosystems

Freshwater bodies in many areas of eastern North America and northern Europe, that today lie in and adjacent to the areas where precipitation is most acid, are threatened by the continued deposition and further expansion of acid precipitation. Many of these bodies of fresh water are poorly buffered and vulnerable to acid inputs. These ecosystems appear destined to suffer greater acidification and loss of fish populations. Equally as serious as damage to fish are the less conspicuous effects of the acidification of fresh water, including changes occurring in communities of aquatic organisms such as microdecomposers, algae, aquatic macrophytes, zooplankton and zoobenthos.

Unpolluted, soft water lakes are generally dilute solutions of Ca and Mg bicarbonate. The bicarbonate system constitutes the main buffering system in the water. Lakes in regions underlain by highly resistant, carbonate-poor rocks have low buffer capacities, and are vulnerable to input of acid precipitation.

Large areas of the U.S. are underlain by granitic rock and are potentially sensitive to acid precipitation, e.g. northeastern U.S. (e.g. Galloway and Cowling, 1977). A major number of the lakes in Scandinavia fall within this category, especially above the postglacial marine limit where the bedrock over large areas is covered by only thin glacial deposits. A continuous supply of acid substances to lakes and streams eventually leads to the depletion and loss of the normal buffer system, pH falls to below 5.0 and sulfate becomes the major anion. Such lakes have only minimal capacity to neutralize additional inputs of acid; and new inputs of acid cause sharp drops in pH (Gjessing et al., 1976).

The acidification of thousands of lakes and rivers in southern Norway and Sweden during the past two decades has been linked to acid from precipitation (Dochinger and Seliga, 1976; Braekke, 1976; Wright and Gjessing, 1976; Almer et al., 1974; Dickson, 1975). In turn, this increased acidity has resulted in the decline of various species of fish, particularly trout and salmon. The fish population in rivers and lakes in 20% of the area of southern Norway have been affected by increasing acidity.

About 10,000 Swedish lakes are estimated to have been acidified to a pH below 6.0 and 5,000 lakes to a pH of less that 5.0 (Dickson, 1975). Along the west coast of Sweden, about 50% of the lakes have pH values of less than

6.0 and pH has decreased as much as 1.8 units since the 1930's. Fish populations have been correspondingly decimated or seriously affected (Dickson, 1975).

The first reports in North America linking lake acidification and the extinction of fish populations to acid precipitation were derived from studies of lakes in the vicinity of Sudbury, Ontario. Populations of lake trout, lake herring, white suckers and other species disappeared rapidly during the 1960's from a group of remote lakes in the LaCloche Mountain region, some 65 km distant from Sudbury. The rapid acidification and fish population extinction in this sensitive region was attributed to the spread of acid deposition from the metal smelters in Sudbury, which were emitting 2.4 million metric tons of sulfur dioxide annually and had recently increased stack height significantly. Increases in acidity of more that one hundredfold in the past decade were observed in some lakes, and of 150 lakes surveyed, 33 were classified as "critically acidic" (pH less that 4.5) and 37 were described as "endangered" (pH 4.5 - 5.5). (Beamish, 1976).

Similar effects have been observed in the Adirondack Mountains in New York. A recent survey found that 51% of the mountain lakes (217 lakes at an elevation above 600 m) have pH values below 5.0; 90% of these lakes contain no fish. In contrast, during the period 1927-37, only 4% of these lakes had a pH under 5.0 or were devoid of fish (Schofield, 1976).

The gradual disappearance of fish usually is attributed to a failure of recruitment, and acid water first affects fish egg and fry. In addition, massive kills of salmon and trout have been observed during snow-melt and after heavy rain. Physiological studies strongly indicate that failure in body salt regulation is the primary cause of fish death in acid water. Therefore, the content of dissolved salts in the water influences the acid tolerance, and field surveys show that the fish populations disappear at a higher pH in lakes with extremely low ion content.

Other evidence indicates that not only are fish affected by acidification, but that a variety of other aquatic organisms in the food web are adversely altered (Dochinger and Seliga, 1976; Braekke, 1976; Hendrey et al., 1976). In general, algal communities in lakes with pH under 6.0 contain fewer species, with a shift toward more acid-tolerant forms. In particular, the chlorophyceae (green algae) are reduced in acid lakes. Some acid lakes and streams contain greater amounts of benthic moss (Sphagnum) and attached algae, and the growth of rooted plants is reduced. There is a tendency toward fewer species of aquatic invertebrates both in the water column and in sediments in acid lakes and streams. The rate of decomposition of organic matter is reduced, with bacteria becoming less dominant relative to fungi. Swedish workers have observed thick fungal felts over large areas of sediments in some acidified lakes. They concluded that decreased decomposition of organic matter on the bottom of lakes, coupled with greater abundance of submerged mosses and fungal mats, reduces nutrient cycling from the sediments. This in turn leads to depletion of nutrients and reduced productivity in acid lakes.

Acid precipitation also causes other changes in lake-water chemistry as well. Elevated concentrations of aluminum, manganese, zinc, cadmium, lead, copper, and nickel have frequently been observed in acidified lakes (Wright and Gjessing, 1976; Beamish, 1976; Dickson, 1975). The abnormally high concentrations are apparently due in part to direct deposition with precipitation as well as increased release (solubility) from the sediments in acidified lakes (Dickson, 1975; Galloway et al., 1976; Galloway and Likens, 1977). These heavy metals may represent a major physiological stress for various aquatic organisms.

Effects of acid precipitation in terrestrial ecosystems

In recent years concern has been expressed that forest growth may also be affected far away from emission sources, where the concentration of acid in air and precipitation is lower than where acute damage and visible symptoms occur. The rate of forest growth has declined in southern Scandinavia and in the northeastern U.S. between 1950 and 1970, but it is not possible to state unequivocally that this decline is caused by acid precipitation (Abrahamsen et al., 1976; Dochinger and Seliga, 1976; Tamm, 1976).

Terrestrial ecosystems are very complex, with numerous living and nonliving components. Since acid precipitation is only one of many environmental stresses, its impact may enhance, be enhanced by, or be swamped by other factors. Recent experiments indicate that acid precipitation can damage foliage; accelerate cuticular erosion; alter responses to associated pathogens, symbionts, and saprophytes; affect the germination of conifer seeds and the establishment of seedlings; affect the availability of nitrogen in the soil, decrease soil respiration; and increase leaching of nutrient ions from the soil (Abrahamsen et al., 1976; Malmer, 1976; Tamm, 1976).

Although many of these factors might be expected to adversely affect tree growth, it has not yet been possible to demonstrate unambiguously decreased tree growth in the field. However, it is possible that acid damage might have been partly offset by the nutritional benefits gained from nitrogen compounds commonly occurring in acid precipitation. Changes already detected in soil processes may as yet be too small to affect plant growth.

Forests are complex. It has been shown that the nature of throughfall (rainfall reaching the forest floor without being intercepted by the crown canopy) and stemflow (rainfall reaching the forest floor by draining down the trunks of trees) is affected differently by different tree species. Thus the composition of "precipitation" reaching soil, possibly affecting soil processes and transfers to freshwater systems, could be influenced by the nature of the tree cover.

Observations of Acid Precipitation in the Western U.S.

Precipitation in much of the west is lower in acidity than is the case in the eastern U.S. Local deposition of acid substance may tend to neutralize excess alkalinity in some western soils. Long-distance transport and deposition, however, will tend to increase deposition of acid substances in both the eastern states and in the west.

In the western United States there is increasing evidence that acidic precipitation exists both in the vicinity of major point sources and also in and near large urban areas. In Pasadena, California measured values of pH in rainfall during portions of 1976 and 1977 show a range of 2.7 to 5.4 with a weighted mean value of 3.9 (Morgan and Liljestrand, pers. comm.). This is a pattern of acidity which is commonplace in the eastern U.S. In the San Francisco region, pH of rainfall has similarly been measured (McColl, pers. comm.) and the indication is that the pH is below the CO_2 equilibrium value of 5.7. McColl's data show that 80% of the samples taken have a measured pH of less than 5.2 with a range of 4.8 to 5.6 and a mean pH of 4.9. The Seattle-Tacoma area has likewise had an acidic precipitation measured at distances from the major SO_2 sources at the Tacoma Smelter and nearby refineries.

SUMMARY

Anthropogenic emissions of sulfur and nitrogen oxides appear to be major precursors of acid-forming anions in acid precipitation. Strong acids have lowered the pH of precipitation in the eastern United States and in much of northern Europe to between 3 and 5.

The most startling effects discovered so far of such air pollutants, have appeared in relatively pristine, remote areas of Norway, Sweden and the eastern U.S. and Canada.

It is virtually impossible to foresee all the ultimate consequences of increasing emissions. The effects on terrestrial ecosystems are often subtle and difficult to document over the short term. Adverse effects on aquatic systems have been reported from several countries.

As we burn ever-greater quantities of fossil fuels, we can expect the impact of these and related air pollutants on sensitive ecosystems to become more serious even though their overall severity still has to be evaluated. Effects of acid precipitation and pollution in general on natural ecosystems should be considered from a broad perspective - not only as to the present but also toward some time in the future when possible effects may be proved beyond doubt to be cumulative and/or become irreversible, as well as intolerable from socioeconomic considerations.

The National Energy Plan calls for continuing increases in the combustion of fossil fuels and consequently for continuing increases in total emissions to the atmosphere. For this reason, it is essential that the United States develop and maintain an adequate national network to measure the amount, chemical form, and geographical distribution of the deposited material and the biological consequences of that deposition in the various regions. Development of such a network should be an integral part of the National Energy Plan.

BIOTIC AND ABIOTIC INTERACTIONS

Reactions of an organism to an air pollutant may be altered by the presence of other pollutants which may alter the degree and incidence of biotic stresses. Air pollutants do not occur alone in the atmosphere; rather they are complex emissions containing a mixture of gases and particles. Combustion of coal, for example, results not only in the emission of large quantities of SO_x and NO_x, but also fluorides and metals, in gaseous and particulate forms; hydrocarbons; carbon dioxide; and a melange of other compounds. Depending upon meteorological considerations, these emissions can merge with those from other point or area sources. When the mixtures come into contact with a receptor, toxic components can alter the normal processes or organisms. One can then ask the following questions: 1) do the individual pollutants in this mixture act on the organism independently or does the presence of one pollutant alter the susceptibility of the host to another? 2) do air pollutants alter the susceptibility of the receptor to pathogens? and 3) do air pollutants alter the suitability of the receptor as a habitat for destructive insects?

Effects on the Plant-Pollutant Interactions

Only limited studies have been conducted on the effects of pollutant mixtures on biological systems. The majority of these studies have been limited to effects (foliar injury, growth and yield) on plant systems (Krause, 1975; Krause and Kaiser, 1977; Reinert et al., 1975). Pollutant mixtures, for example, SO_2+NO_2; SO_2+O_3; SO_2+HF; $\overline{NO_2}$+$\overline{HF}$ and SO_2+ metals, have been studied. Three general types of plant effects have been reported: Synergism, where the effects of the pollutant mixture were greater than would be expected from the sum of the effects of individual pollutants; independence, where the pollutants acted separately to produce their effect; and antagonism, where the presence of one compound decreased the phytotoxicity of another.

The Magnitude of the Effect

The present data are inadequate to indicate the significance of pollutant interactions in terms of their effects on foliar injury, growth, and yield. However, mixtures of pollutants do exist in the environment and some level of effects probably occurs. With the projected increases in SO_x, NO_x (Figure 1) and probably HF, the likelihood of effects from these pollutant mixtures will increase. It is also likely that there will be increased interaction of effluents from power generation and pollutants from urban sources which could increase the biological effects. But based on the limited data base, the magnitude of these effects cannot be estimated.

Diseases

The effect of air pollutants in the interactions among compartments of the biotic environment has been recognized as a significant consequence of air pollution and may be expressed, with respect to plant diseases, as making the host a more or less suitable habitat for the pathogen.

Information on the effects of pollutants on the plant-pathogen relationship has been based upon three types of evidence: 1) field observations; 2) experimental exposure of the host-pathogen system; and 3) studies of the toxicity of the pollutant and the pathogen alone. Field and laboratory observations suggest that, in general, the presence of SO_2 results in a decrease in the incidence and severity of fungus diseases (Heagle, 1973; Weinstein et al., 1975). In some cases, where a fungus infection already exists, susceptibility to SO_2 may be enhanced (Chiba and Tanaka, 1968).

In the case of ozone, its effect on plant pathogens depends upon the time of exposure to the pollutant in relation to the time of inoculation. In general, the presence of ozone-induced and severity of certain diseases, with exposure to ozone after inoculation often results in a lesser incidence and severity of disease. A considerable body of evidence also exists that suggests that infection by a fungus confers a measure of resistance to subsequent ozone exposure (Treshow, 1975; Heagle, 1973; Weber, 1977). Inhibition of nematodes in soybean roots has also been reported after exposure of the plant to O_3 alone or in combination with SO_2 (Weber, 1977).

Insects

In several geographical areas, an increased incidence of insects in natural vegetation is associated with injury due to air pollutants. The kind of effect that occurs in the field depends upon the characteristics of the pollutant(s), conditions of exposure, environment, and species of plant and insect. There are two hypotheses to explain this effect: 1) pollutants exert a direct toxic effect on the insects and 2) pollutants change the susceptibility or attractiveness of the plant to the insect. Oxidant pollution injury has been shown to predispose ponderosa pine to invasion by western and mountain pine beetles (NAS, 1977). Doane (1915) described many insects in the plume zone of the Selby smelter in California but did not relate this incidence and severity to the occurrence of SO_2.

The presence of airborne particulates is known to be related to the incidence of outbreaks of certain destructive insects (Bartlett, 1951; Edmonds, 1973).

Areas of Uncertainty

The mechanisms by which air pollution alters the relationships among plants, insects, and pathogens is not understood except perhaps in the case of invasion by bark beetles of trees weakened by oxidant pollution.

One cannot assign any degree of confidence to interpretations of the many and varied observations made in the field and under controlled laboratory conditions until there is an understanding of certain mechanisms, some of which are given below.

1) How is the response of an organism altered by the imposition of two or more pollutants?

2) How does an air pollutant applied to the plant influence the relationships between plant roots, and symbiotic microflora, pathogenic microflora, or microfauna?

3) How does an air pollutant alter the plant as a less or more suitable host for destructive insects?

4) How do air pollutants alter the relationship between destructive insects and their parasites and predators?

5) How do air pollutants alter the plant as a host for fungal, bacterial, viral, and mycoplasmal pathogens?

ECOSYSTEM IMPACTS

Natural terrestrial ecosystems are classified in terms of their dominant life form, namely, deciduous and coniferous forest, chaparral, desert, grassland and tundra, etc. The relationship of the extent of 10 natural forest ecosystems in the United States to an estimate of the meteorological potential for air pollution episodes has been suggested in a recent review by Miller and McBride (1975). See figure 4.

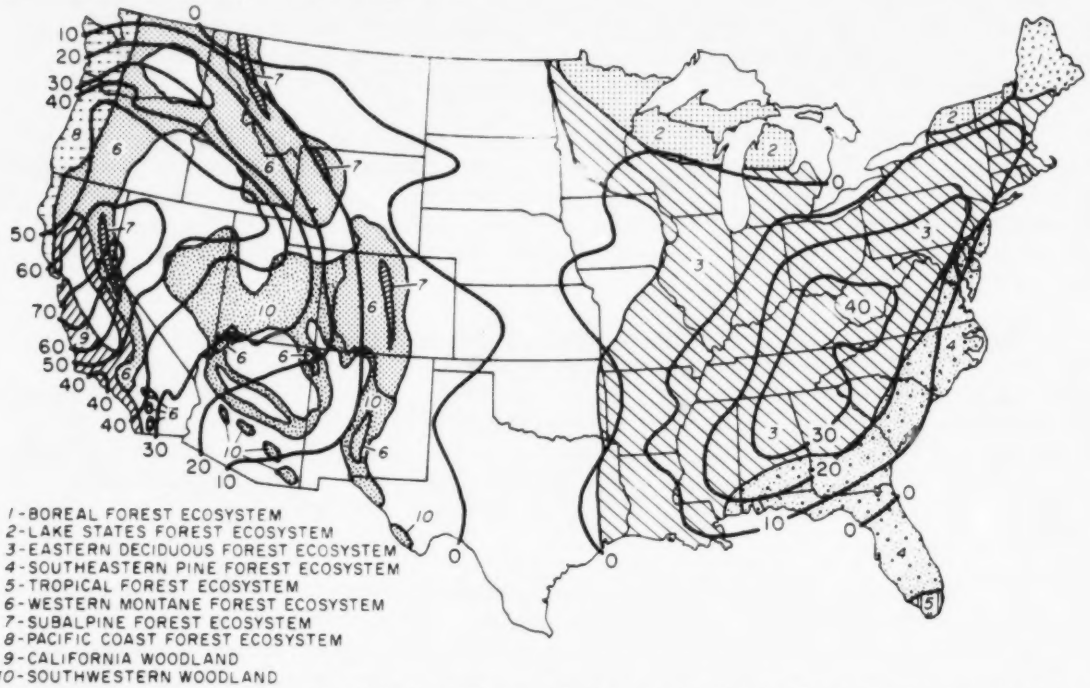

Figure 4. Isopleths of total numbers of forecast-days of high meteorological potential in a five year period (Holzworth, 1972) compared with boundaries of forest ecosystems.

The unshaded intervening areas on the map include the other natural ecosystems (grassland, desert and chaparral) and extensive areas classed as agroecosystems. Aquatic ecosystems, including rivers, lakes, marshes, and coastal marine ecosystems are distributed within and around each of the terrestrial systems.

The significance of the overall effects of sulfur oxides, photochemical oxidants, particulates, and acid precipitation on these terrestrial and aquatic ecosystems varies widely, depending both on the regional load of emissions and the sensitivity of the plant and animal components of each system. Thus the pattern of projected increases in some emissions by 1985, and decreases in others can be expected to yield major differences in regional ecosystems.

Plant Population Responses to Lowered Photosynthesis Rates

The most obvious effect of chronic pollutant exposure in a natural ecosystem flows from the reduced growth rates of sensitive species, including some dominants of the primary producer level (Woodwell, 1970; National Academy of Sciences, 1977). The latter report reviews known and anticipated direct and indirect effects of oxidants on populations at the producer, consumer and decomposer trophic levels. A summary report for studies in the San Bernardino National Forest was sponsored by the Environmental Protection Agency (Miller et al., 1977) which details the impact of oxidants on a western forested ecosystem.

The pollutants described in the previous sections generally constitute new stresses to which existing species are not pre-adapted. As the pollution-tolerant species increase their importance in the ecosystem, changes in succession and diversity will result (Loucks, 1972; Glass and Tingey, 1975) as well as altered rates of biogeochemical cycling and altered rates of energy retention.

The overall result of these processes can be summarized briefly as follows: 1) Biotic composition of the system will continue to change until the mix of species best adapted to the new stress sources become the dominants. 2) Associated with these changes are shifts to (i) less carbon found in biomass; (ii) larger R/P for the system as a whole and for the major species; (iii) fewer species, and fewer individuals of species of relatively large biomass; (iv) more short-lived, highly productive (r-adapted or weedy) species (Odum, 1969).

Effects on the primary producer component can have other consequences. Studies by Schwartz et al. (1977) have shown a 10 to 20% decrease in crude protein matter digestibility levels in western wheatgrass exposed to chronic low levels of SO_2. Western wheatgrass is one of the dominant forage grasses in the western plains and a significant change in protein or digestibility can have important consequences for consumer species (including domestic livestock) dependent on it.

The overall significance of the plant injury process is usually interpreted in terms of effects on the ecosystem's standing stock, i.e., the system's "free goods" (Westman, 1977). In effect, chronic injury may constrain: 1) "the direct harvest of marketable products", or 2) "the use and appreciation of ecosystems for recreation, aesthetic enjoyment, and study" (Westman, 1977).

Changes in Biogeochemical Cycles

The principal perturbation of biogeochemical cycles steming from increased coal utilization is through the induction of acid rain. These effects have been reviewed in detail in a previous section and are only recapitulated briefly here. Analyses of forest growth in southern Sweden from 1896 to 1965 showed a 2 to 7% decrease in growth between 1950 and 1965. Johnsson and Sundberg (1972) "found no good reason for attributing (this) reduction in growth to any cause other than acidification." In addition to the effects on growth, the acidification can inhibit decomposition of litter from the forest floor (Abrahamsen et al., 1976).

A major influence, therefore, is that nutrients can be released due to a decrease in standing crop of dead as well as living organic matter. Accelerated leaching due to acid rain could remove the nutrients from the nutrient pool to be bound in sediments of lakes and ocean. The smaller nutrient pool could ultimately constrain the re-development of previous biomass levels (Woodwell, 1970).

In addition to acid rain effects, however, biogeochemical cycles are greatly modified by the direct addition of nutrients, especially nitrogen (Likens et al., 1977). The balance between increased leaching and increased nitrogen additions will vary regionally as the magnitude of the acid rain and NO_x emissions vary.

Impacts on Terrestrial and Aquatic Ecosystem Carbon Flux

Nitrogen and sulfur oxide deposition from air pollution is presently of sufficient magnitude to act as a major nutrient input and modifier of nutrient cycling in terrestrial ecosystems. Depending on geographic location roughly half of the inputs are dry deposited and half wet deposited. Possible long-term responses may range from fertilization to increased successional rates due to removal of nutrient constraints. Specific impacts are likely to be affected to the extent that the system is nutrient deficient or the cycle is modified to produce serious nutrient losses. The response observed at specific sites will depend on local characteristics of the soils.

For aquatic ecosystems, the following step damage functions may be used to describe general impacts associated with lake acidification (G.R. Hendrey personal communication);

1. Long-term changes of less than 0.5 pH unit in the range 8.0 - 6.0 are likely to alter the biotic composition of fresh waters to some degree. Probably no significant changes.

2. A decrease of 0.5 - 0.9 pH unit in the range 8.0 - 6.0 may cause detectable alterations in community composition. Productivity of competing species will vary. Some species will be eliminated.

3. Decreasing pH from 6.0 to 5.5 will cause a reduction of species numbers. Among remaining species, significant alterations in ability to withstand stress will occur.

4. Below pH 5.0 - many species will be eliminated. Molluscs, amphipods, most mayfly and many stonefly species. Several tolerant invertebrates will become abundant. (Siolis, notoneclan, corixials, chironomis).

5. Below pH 4.5, all of the above changes are exacerbated and all fishes are eliminated.

Reductions in pH reduce bacterial activities and bacterial abundance/unit organic matter. Also, phycomycetes replace bacteria as the dominant saprotrophs. Mineralization is greatly retarded and organic debris accumulates.

An additional influence of dry and wet deposition of oxides of sulfur and nitrogen on aquatic ecosystems is the eutrophication that can result from the input of these materials as nutrients in both freshwater and marine systems. Most freshwater ecosystems are phosphorus limited, but some are nitrogen limited, at least seasonally (Miller, et al., 1974; Schindler, 1977). Many marine and estuarine ecosystems are nitrogen limited (Ryther and Dunstan, 1971). Increased deposition of nitrogen due to fossil fuel combustion will generally not have large effects on lake productivity unless this is accompanied by increased phosphorus loading or unless the lake is nitrogen limited. However, because of ever increasing human population and consequent increases in phosphorus output from sewage treatment plants, the potential for nitrogen limited bodies of water will probably be greater in the future than it is now. Oligotrophic lakes are likely to be the most vulnerable to nutrient inputs from wet and dry deposition per unit of loading than are more eutrophic lakes and can, therefore, be considered more sensitive.

Discussion of Overall Effects

The long-term consequences of stresses on ecosystems have been briefly summarized by Loucks (1972). Analyses have now been completed that show the results of whole-system upsets and the types of evidence needed to document them. Harrison et al. (1970) used systems methods and the results of studies of DDT effects on populations to examine whole ecosystem response characteristics. These upsets may cause large-scale changes in population levels of certain species and their replacement by other species, or their complete elimination. Both studies indicate a trend toward simplification of the system when placed under the stress of a contaminant, but the long-range effects of such simplification cannot yet be fully evaluated.

Further work is needed on examples of potential ecosystem perturbations on the role of native species diversity in maintaining the stability of these systems and of the long-range effects of the trend toward simplification.

Although lichens and bryophytes have been regarded as very sensitive components, initial effects may be manifest in the dominant and codominant

trees as the canopy starts to decline in growth and vigor owing to its exposure to a greater pollution flux. When exposures of sufficient magnitude have occurred, successive layers of vegetation are removed resulting in its most drastic effect, barren soil and erosion (Rennie and Halstead, 1977).

An approximate indication of the level of knowledge concerning important processes common to the ecosystems shown in Figure 4 is shown in Table 1.

TABLE 1. ESTIMATES OF THE PRESENT BREADTH OF KNOWLEDGE CONCERNING MAINLY POLLUTANT INDUCED PERTURBATIONS IN ECOSYSTEMS.

Changes	Ecosystem				
		Forest			
	Grassland	Conifer	Deciduous	Chaparral	Desert
Primary Productivity	+	++	+	?	?
Reproductive Capacity	+	+	0	?	?
Host-Parasite Relationships	0	++	+	0	0
Litter Decay	0	+	0	0	0
Nutrient Storage	+	+	++	0	0
Successional Patterns	0	+	+	?	?
Animal Morbidity and Mortality	0	?	0	0	0

0 = no knowledge of effects
? = probable effects
+ = limited knowledge
++ = substantial knowledge

Single and double plus signs have been assigned where first approximation simulation models are now available to study the effects of chronic oxidant exposure. Obviously, gaps in the table are more profound than apparent knowledge.

During the last 20 to 30 years, the problem of elevated photochemical oxidant concentrations has become a new stress in several ecosystems particularly in the Southwest. Plant and animal communities in natural ecosystems have evolved over thousands of years in the presence of periodic stresses (droughts) and catastrophic events (fire). Over this time span, certain species have developed successful mechanisms for survival so that successional development toward a stable climax community may resume its course. In a western mixed conifer forest ecosystem, when oxidant stress is superimposed on the natural stresses (drought and fire) the course of developemnt in some forest stands appears to be a rapid decline in the population of the most fire tolerant species, ponderosa and Jeffrey pines. It is possible that repeated fire events could inhibit the recovery of the stands because the remaining oxidant tolerant but fire sensitive species may be nearly eliminated in just a few decades. The former forest cover may yield to a less desirable grass or shrub cover.

It is clear that oxidant pollution can affect ecosystems by adversely influencing the nitrogen fixation process and in other ways as well (Taylor and Miller, 1973; Miller and McBride, 1975). SO_x, as shown above, also influences nitrogen fixation but, as with oxidant pollution, affects a number of other ecosystem processes in addition.

Oxidant air pollution and SO_x emissions are the major air pollutants from transportation-oriented and stationary sources, respectively. In the western United States, the preponderant air pollution problem is associated with oxidants. In the eastern United States, the dominating forms of air pollution are SO_x, NO_x, and oxidant. Ecosystems in all parts of the country are being affected in one way or another by these two types of air pollution emissions due to long distance transport. The mechanisms whereby air pollutants exert an impact on natural and man-made or managed ecosystems are numerous, but most mechanisms are related to photosynthesis or changing community structure and at this point are too poorly understood to make definitive forecasts of consequences of increased coal utilization.

EFFECTS ON MATERIALS

The damaging effects of air pollution on a wide variety of materials have been recognized for many decades. Sulfur oxides, nitrogen oxides, and particulate matter are the pollutants emitted by coal-burning facilities and known to damage materials. Estimates indicate that sulfur oxides cause about 35 percent of the total damage to materials. For example, atmospheric SO_2 and its derivatives can 1) accelerate the corrosion rates of most ferrous metals and some non-ferrous metals, mainly zinc galvanized products; 2) reduce the durability of exterior paints containing calcium carbonate fillers; 3) attack and cause pitting in limestone, marble, mortar, and concrete building materials; 4) deteriorate cellulosic and some man-made textile fibers; 5) produce color changes in some textile dyes; and 6) cause paper and leather products to lose strength, a potentially serious problem for some urban museums and libraries.

Oxides of nitrogen (mainly NO_2) and subsequent reaction products (nitric acid and nitrates) can cause certain textile dyes to fade and some textile additives to yellow, deteriorate cellulosic fabrics, and accelerate stress corrosion of certain metal alloys. In assessing NO_x damage to materials, however, one must not lose sight of the role that NO_x plays in the photochemical formation and buildup of ozone and other oxidants, as well as in the photo-oxidation of SO_2 aerosols. These photochemical reaction products are believed to cause more damage than those associated directly with NO_x.

The most obvious effect of airborne particles is the soiling of property-buildings, homes, automobiles, etc. Soiling generally produces a need for more frequent cleaning and/or painting, thus resulting in an economic burden. Certain kinds of particles can also cause direct chemical damage to materials because of corrosive chemicals sorbed by the particulates. Sulfuric acid in the form of liquid aerosols or attached to fly ash particulates is well known to cause damage to paint and structural materials in the vicinity of some oil and coal-fired power plants.

Meteorological factors strongly influence the rate that pollutants damage materials. Important factors are relative humidity, temperature, rainfall, sunlight and air movement. Relative humidity is the most important; damage can be significantly greater in more humid (> 60% RH) areas than in drier areas.

In attempting to estimate the cost of the material damage attributed to air pollution, economists recognize the complexities of developing meaningful cost figures mainly because information gaps exist in two broad areas: (1) inadequate damage function or effects data on economically important material

receptors, and (2) a general lack of information on the spatial and temporal distribution of pollutant levels and receptor populations. Nevertheless, a number of gross damage estimates have been made. Comparative studies made in polluted and nonpolluted areas have highlighted the corrosion effects of sulfur dioxide on materials. The role of acid precipitation has not yet been clarified. In 1970 the annual costs per person of damage ascribable to the main atmospheric sulfur pollutants were estimated to be about $7 and $4 in the the USA and Sweden, respectively (Kucera, 1976). The most recent estimate for material losses in 1970 (excluding soiling costs) was $1.7 billion per year. This figure was largely arrived at by summing previous damage estimates for specific material categories. A breakdown of this figure shows that $0.6 billion was attributed to SO_x, $0.9 billion to the combined effect of NO_x and oxidants, and $0.2 billion to particulate matter. Paint and zinc (galvanized products), mainly because of the large total surface area exposed to the outdoors, are the most vulnerable materials from an economic standpoint. However, certain test methods such as painted panels and losses of textiles may be excessive because physical damage functions are in error. Therefore, economic losses may not be as high as the $1.7 billion.

Because of the information gaps, reliable estimates of the incremental economic impact on materials resulting from increased consumption of coal specifically is not possible. Gillette (who has written an assessment of the costs of SO_2 damage to materials) estimates that a 10 percent increase in overall levels of urban air pollution would probably increase the economic material loss by 20-30 percent, whereas a 25 percent increase in air pollution would probably more than double these losses.

Gillette (1975) points out, however, that any estimate must be qualified because materials have pollutant threshold levels below which economic damage does not occur. Threshold levels are based on the premise that, while some materials may have an infinite life under ideal environmental conditions, manufactured products have a finite expected service life mainly because of obsolescence. If the actual use life of a product is less than its normal or ordinary use life because of damage by pollution, an economic loss has occurred. These losses are reflected in the market in the form of increased expenditures for replacement or maintenance.

There are four areas in the economic functions which would add an error margin of possibly a factor of 2 to 3 to existing estimates.

1. Present techniques do not reflect the introduction into the market place of highly resistant materials (acrylics, other polymers, self-weathering steels, aluminum clad steels).

2. Estimates assume that galvanized steels are left uncoated. This is not the case. Therefore, the amount of surface which is highly suspectible to damage is greatly reduced.

3. Many materials have use lives which are so short, that pollutant damage has little effect on them. In some cases, the item is used regardless of rusting as long as it functions.

4. Present loss estimates assume that a substitute material costs more than the original material, the cost differential is attributed to pollution. Due to the increasing cost of labor, materials are being sought which reduce maintenance, and at the same time are resistant to pollutant attack. Savings realized from lower maintenance should be deducted from the costs charged to pollution.

EFFECTS ON VISIBILITY

Visibility reduction is probably the most immediate and obvious effect of air pollution, and is the best understood and most easily quantified effect of pollution by fine particulate. Visibility can be simply defined as the farthest distance at which an unaided human eye can see a black object located at the horizon. In an atmosphere completely free of particulate contaminants, the visibility is limited to about 200 km due to the scattering of light by gases composing the atmosphere. In actuality aerosols exist throughout the atmosphere and scatter additional light, reducing the visibility in some places to appreciably shorter distances. Visibilities in the western United States often approach the limit of 200 km, and 100 km is attained regularly in noncloudy, rural settings. Public Law 95-95 requiring nondegradation of visibility in federal class I regions has elevated the importance of this problem, mainly to protect scenic vistas. PL 95-95 also protects class I areas from further ecological damage as well.

Our knowledge of the causes and characteristics of visibility reducing aerosols has increased greatly in the past ten years. One of the most significant realizations has been that aerosols between the sizes of 0.1 and 1.0 μm are responsible for a majority of the visibility reduction (Waggoner and Charlson, 1977). This fact results from two phenomena: first, particles smaller than 0.1 μm, though present in very high number, are very inefficient in scattering light and thus contribute very little to visibility reduction; second, particles larger than about 1.0 μm, are both inefficient in scattering light, and usually exist in quite small numbers and contribute to only a small fraction of the visibility reduction (except in dust storms and fog).

Thousands of measurements of atmospheric aerosol size distributions have shown that on a mass, volume, or surface basis the aerosol exists in up to three distinct size modes: nuclei (< 0.1 μm diameter), accumulation (0.1 μm to 2.0 μm), and coarse particle (> 2.0 μm). The number of modes, mean diameter, and the mass, volume, or surface in each mode depend upon the source and history of the aerosol. Coarse particles consist of windblown dust, fly ash, plant parts, powders from any grinding or pulverizing operation and activated clouds and fog drops. They are removed by sedimentation and precipitation and have little interaction with fine particles. The nuclei mode originates mainly from the condensation of hot, supersaturated vapors during combustion processes. It loses particles primarily by coagulation to larger sizes. Accumulation mode particles are formed by the coagulation of smaller particles, by the condensation of vapors onto existing smaller particles, and by the chemical reaction of vapors with existing smaller particles. Coagulation is not a major removal mechanism for aerosols of accumulation mode size because of the rapid decrease on number concentration in that size range (Higgins, 1977).

Because aerosols in the 0.1 to 1.0 μm range are responsible for most visibility reduction and because fine particles tend to accumulate in that same range, it follows that to control visibility reduction one must control the entry of particles into the accumulation mode. The composition of accumulation mode aerosol has therefore been the subject of much recent research. Current evidence is that sulfate often accounts for about half of the accumulation mode mass, for example in industrial regions. The most direct evidence has been measurement of sulfur and sulfate by x-ray fluorescence and wet-chemical methods (Stevens, 1978). Supporting evidence comes from the humidified nephelometer work of Waggoner, (Waggoner and Charlson, 1977) in which water vapor is added to ambient aerosol and the light scattering is measured; increases in light scattering characteristic of various sulfate aerosols were found during field studies in Missouri, Michigan, and Arkansas (Weiss, 1977). Further evidence is given by the regression analyses of Trijonis (Trijonis and Yuan, 1977), Barone (Barone, In press), and Cass (Cass, 1976). Though regressions do not necessarily imply cause-and-effect relationships, a lack of correlation between visibility reduction and sulfate would imply no cause-and-effect; in these studies definite correlations exist. Another study implicating the visibility reduction sulfate link is Trijonis' analysis (Trijonis and Yuan, 1977) of the southwest U.S. copper strike of 1967-1968. During the strike copper smelters were shut down for nine months and resulting NASN sulfate levels were 38 to 76% below seasonal averages of surrounding years. At the same time airport visibility measurements improved by 5 to 25%.

It is significant that simultaneous, direct measurements of optical and concentration data agree, with a ratio of sulfate mass concentration to extinctions of ca 0.1 g/m^2. The simultaneous agreement of these two approaches and with computations by Waggoner et al., 1976 lend credibility to all these approaches.

The origin of anthropogenic sulfate in the accumulation mode is sulfur dioxide which has been converted by any of several possible atmospheric reactions. Recent regional-scale field studies by the U.S.E.P.A. indicate that SO_2 to sulfate conversion rates in power plant plumes can account for significant sulfate formation, as well as transport many hundreds of kilometers downwind (Wilson, 1977). Thus, even if large sources of sulfur dioxide were to be located in rural areas, urban as well as rural sulfate levels could be expected to increase, accompanied by reduced visibility.

In regions prone to high relative humidities three phenomena may aggravate the sulfur dioxide-sulfate-visibility relationship. First, there is an increased conversion rate of sulfur dioxide to sulfate, (Eggleton and Cox, In press; Beilke and Gravenhorst, In press) thus, more net sulfate may be formed due to the reduced opportunity for SO_2 removal by dry deposition. Second, the visibility reduction of a sulfate aerosol always increases as the relative humidity increases because the particles grow in size (Covert, 1972). The visibiltiy reduction of some sulfate species (acid sulfates, such as sulfuric acid) increases smoothly with increasing humidity; other species (e.g. ammonium sulfate) exhibit sudden particle size growth at certain humidities and double or triple their visibility reduction above those humidities. Third,

when particles of certain sulfate species (e.g. ammonium sulfate) have grown suddenly because of sufficiently high humidity, they do not return to their former size as the humidity falls below that same critical value (Yamamoto, 1975); thus, the visibility reduction persists until a lower humidity is reached. However, the frequent existence of RH < 70% (below 70% these effects become quantitatively unimportant) in areas such as most of the western half of the U.S. allows useful simplification. Even at 80% RH, the effect is only about a factor of two above low RH values.

The impact of increased coal usage on visibility thus depends on the control of sulfur dioxide emissions. If sulfur dioxide emissions cannot be kept at present-day values, rural and urban sulfate levels can be expected to increase, with a further visibility reduction. The effects of further visibility reduction fall into two broad categories: social-political effects and climate modification. Social-political effects range from simple citizen dissatisfaction to decrease in revenue and property values in area of scenic attraction. Potential climatological effects include the reduction of solar radiation for photosynthesis and energy production, heating or cooling of the atmosphere resulting in changes in the length of growing seasons, and changing precipitation levels (Higgins, 1977).

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